Post-fire Recovery of Wyoming Big Sagebrush Shrub-steppe in Central and Southeast Montana

Prepared for:

United States Department of the Interior
Bureau of Land Management
State Office

Prepared by:

Stephen V. Cooper, Peter Lesica and Gregory M. Kudray

Montana Natural Heritage Program
a cooperative program of the
Montana State Library and the University of Montana

December 2007

MONTANA
Natural Heritage
Program

Post-fire Recovery of Wyoming Big Sagebrush Shrub-steppe in Central and Southeast Montana

Prepared for:

United States Department of the Interior
Bureau of Land Management
State Office

Agreement Number:

ESA010009 Task Order #29

Prepared by:

Stephen V. Cooper, Peter Lesica and Gregory M. Kudray

P.O. Box 201800 • 1515 East Sixth Avenue • Helena, MT 59620-1800 • 406-444-5354

This document should be cited as follows:

Stephen V. Cooper, Peter Lesica and Gregory M. Kudray. 2007. Post-fire Recovery of Wyoming Big Sagebrush Shrub-steppe in Central and Southeast Montana. Report to the United States Department of the Interior, Bureau of Land Management, State Office. Montana Natural Heritage Program, Helena, Montana. 16 pp. plus appendices.

Executive Summary

Sagebrush is a widespread habitat throughout our study area and a number of species including Greater Sage-grouse, pronghorn, Brewer's Sparrow, Sage Sparrow, Sage Thrasher and sagebrush vole are sagebrush dependent, at least at some stage of their life cycles. Fire constitutes an important driver in structuring sagebrush ecosystems; past investigations have established that the response of the big sagebrush component (*Artemisia tridentata* Nutt.) varies according to subspecies. In an earlier study in southwestern Montana we statistically determined that recovery of mountain big sagebrush (*A. t.* ssp. *vaseyana* [Rydb.] Beetle) cover occurred in slightly more than 30 years, however the minimal data for Wyoming big sagebrush (*A. t.* ssp. *wyomingensis* Beetle & Young), indicated a much longer recovery period (Lesica et al. 2005). In this study we used the same sampling protocol at 24 burned-unburned paired sites in central and southeastern Montana where Wyoming Big Sagebrush is the dominant big sagebrush taxon and the accompanying flora is more closely allied with the Great Plains than the Intermountain West.

Prescribed burns and wildfires typically result in the complete mortality of Wyoming big sagebrush. We found that Wyoming big sagebrush recovers very slowly from both types of burns at all sites, even those with relatively moist conditions. Full recovery to pre-burn sagebrush canopy cover conditions will take well over 100 years. The median time since fire was 22 years and ranged from 4 to 67 years. We found no Wyoming big sagebrush canopy cover recovery for 17 of the 24 sites after burning had occurred and the oldest burn was only 8% recovered. Livestock grazing does not seem to be casual as the only site without livestock grazing for the entire period after burning had no canopy recovery in 25 years. Burned plots were located near unburned areas to ensure that a seed source was relatively available since Wyoming big sage is known to lack a soil seed bank.

Perennial and annual grass cover increased after burning, however virtually all of the 11% increase in annual grass is from field brome (*Bromus arvensis*, formerly Japanese brome, *Bromus japonicus*), regarded as a weed with negative habitat and livestock value. Perennial grass cover increased 27% and 20% followed prescribed fire and wildfire, respectively. Western wheatgrass (*Pascopyrum smithii*) increased by 17% and accounted for most of the perennial grass increase. These increases did not decline with time since burning, which may be explained by the lack of the competitive influence of sagebrush recovery. There was no change after burning in overall forb cover or the numbers of forbs of the Cichorieae Tribe of the Asteraceae family. The Cichorieae tribe forbs are important for successful Greater Sage-grouse brood rearing. Plant species richness significantly declined in burned plots compared to their unburned control plots.

Our findings of extremely slow Wyoming big sagebrush recovery after fire are similar to the other research in the area (Eichhorn and Watts 1984) and also supports findings by Baker (2007) that fire rotations for this subspecies are about 100 – 240 years.

The slow Wyoming big sagebrush recovery and the increase in the weedy annual grass field brome suggests that managers concerned about Greater Sage-grouse and other sage-dependent species should be extremely cautious with prescribed burns and wildfires in this region. Burns may essentially eliminate sagebrush habitat, increase weedy annual grass cover, reduce species richness, and could take a century or more for recovery to pre-burn sagebrush cover conditions.

Acknowledgements

We especially thank Nora Taylor of the BLM for her generous support in making this project possible. Louise deMontigny and Eric Lepisto of the Miles City Field Office, Bureau of Land Management were instrumental in locating southeastern Montana study sites, as was Jeff DiBendetto of the Custer National Forest (Billings). Michael Stops, Chief Ranger Little Bighorn Battlefield National Monument, graciously permitted sampling at this historical site. Larry Eichhorn (Lewistown, retired BLM range conservationist) was generous with his time, helping us relocate his original sagebrush succession study sites and dispensing informed commentary on the study. Tom Stivers of Montana Fish, Wildlife and Parks shared his knowledge of central Montana sage-steppe fire history crucial to our locating several sampling sites. U. S. Fish and Wildlife personnel at the Charles M. Russell National Wildlife Refuge, especially Bill Berg, Bob Skinner and Joann Dullum, provided location information on past refuge fires. We thank the several ranchers who granted access to their lands. Our sincere appreciation to Lisa Wilson and David Salazar who served as field assistants, admirably discharging their duties and providing informed camaraderie. Coburn Currier's suggestions contributed to manuscript organization and readability before he formatted it to MTNHP specifications.

Table of Contents

List of Figures

List of Tables

Introduction

Sagebrush steppe is a dominant vegetation type in the Great Basin and Intermountain Region of western North America but it is also important in portions of the Northern Great Plains where agriculture (cereal grains) and mixed-grass prairie now dominate. Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) dominated vegetation is an important component of the semiarid landscapes east of the Rocky Mountains stretching from Wyoming through Montana to just south of the border with Canada; it is also found in westernmost North Dakota. Throughout southeast Montana Wyoming big sagebrush is the only subspecies of *A. tridentata* present, usually on fine textured soils; the only other large shrubby sagebrush present in this region is silver sagebrush (*A. cana*), found on drainage terraces and sandy substrates. Physiognomy of Wyoming big sagebrush stands in the Northern Great Plains differs from the Intermountain Region in that the undergrowth is dominated by rhizomatous grasses as opposed to tussock-forming grasses. Also influencing stand physiognomy are two notable clines in Wyoming big sagebrush size presumed to reflect available soil moisture; one of increasing plant height from 1) south to north and 2) from lower to higher elevation. Mountain big sagebrush (*A. t.* ssp. *vaseyana*) is also found within the study area to a limited extent; it occurs at lower treeline and in mountain parklands of the isolated mountain ranges of central Montana.

Fire was instrumental in structuring presettlement sagebrush ecosystems, generating a mosaic of stands of different size in various seral stages (West 2000). Fire, even of low intensity, does not thin or lower sagebrush density by killing some fraction of sagebrush plants throughout a stand, rather it is stand-replacing because mortality is complete when flames reach sagebrush (Baker 2007). Conserving native species diversity likely requires maintaining a comparable mosaic. Greater Sage-Grouse (*Centrocercus urophasianus*), for example, require barren habitats for leks, relatively dense stands of medium height for nesting (Klebenow 1969, Wallestad and Pyrah 1974, Aldridge and Brigham 2002), open stands for brood raising (Klebenow 1973, Wallestad 1971), and full-canopied tall stands for wintering (Eng and Schladweiler 1972). Greater Sage-Grouse populations apparently can be constrained by the loss of any one of these structural types (Connelly et al. 2000, Roscoe 2002). Antelope, Brewer's Sparrow, Sage Sparrow, Sage Thrasher and sagebrush vole are also sagebrush dependent, at least at some stage of their life cycles.

Management strategies that promote the conservation of all sagebrush steppe-dependent species are currently being formulated, and prescribed fire has been proposed as a method to control the density of big sagebrush stands (Klebenow 1973; Pyle and Crawford 1996). However, as post-fire succession proceeds from immediate post-treatment to mature structure, we only have limited knowledge of changes in sagebrush cover, height, associated vegetation and other characteristics. Though considered a climax-dominant species, evidence suggests that big sagebrush burning response varies according to subspecies and may require many years for post-fire re-establishment (Baker 2007). Wyoming big sagebrush, although highly variable in response (Walhof 1997, Wambolt et al. 2001, Watts and Wambolt 1996), has almost no recovery for 30 years (Wambolt and Payne 1986, Eichhorn and Watts 1984) and generally requires at least 50 years to attain a density equal to that of the unburned control (Baker 2007, Colket 2003). With the lone exception of the Eichhorn and Watts (1984) study in central Montana's Missouri River Breaks, none of these studies were conducted in a Great Plains environment. The ecological dynamics and habitat characteristics of these sagebrush communities are almost certainly strongly influenced by their age (size) structure. Landscape scale comprehensive management of sagebrush cannot be achieved without understanding how structural and compositional components change with time since disturbance.

The purpose of this study was to describe and substantiate the change in sagebrush and associated vegetation after fire in the Northern Great Plains of eastern and central Montana. We documented changes in shrub height, cover and size-class

distribution by sampling numerous stands of various post-fire ages and asked whether recovery differed by ignition source (wildfires versus prescribed burns).

Study Area

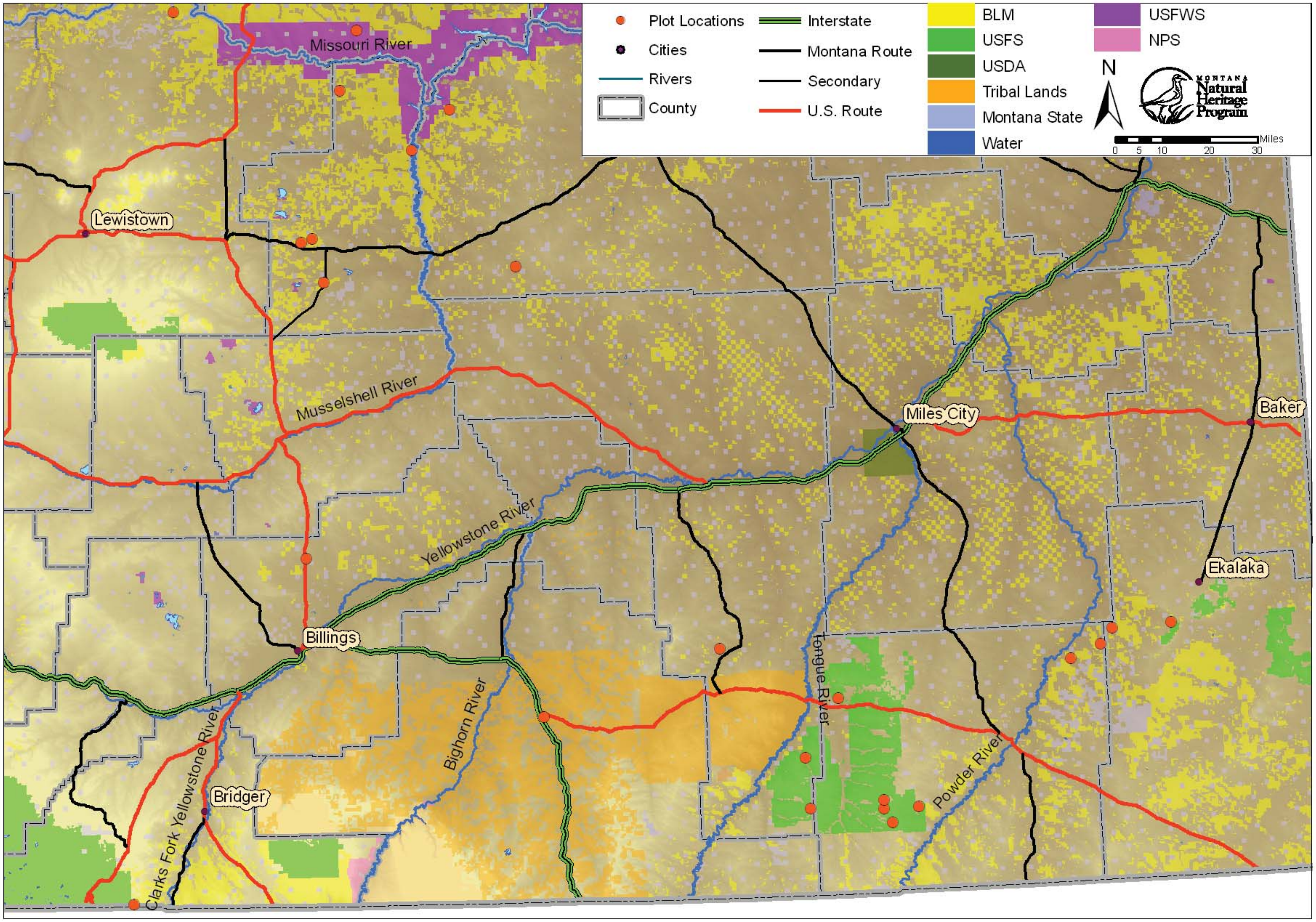
Sampling was conducted over a broad swath of eastern Montana (Figure 1) from a westernmost site within the Bighorn Basin Section, Bighorn Intermountain Basin Subsection (342Ad, Bailey 1995, Nesser et al. 1997) to the eastern-most Section-Subsection, Northwestern Great Plains, Pierre Shale Plains (331Fc). However, most of the sampling occurred in the Northwestern Glaciated Plains Section (within the Montana Glaciated Plains [331Dh] and the Missouri River Breaks [331Df] Subsections) and the Powder River Basin Section (within the Montana Shale Plains [331Gb], the Montana Sedimentary Plains [331Ge], and the Powder River Basin/Breaks/Scoria Hills [331Gc]. All these units, with the exception of 342Ad, occur within the Great Plains-Palouse Dry Steppe Province (Bailey 1995).

Steppe and shrub-steppe vegetation is characteristically associated with semi-arid climatic regimes with an annual precipitation from 250 to 500 mm (10 – 20 inches). The mixed-grass prairie and shrub-steppe results from the relatively low annual precipitation, which according to the DAYMET model (Thornton et al. 1997) varies from 274 mm (10.8 in., vicinity of confluence of Alkali Creek with Musselshell River) to 415 mm (16.35 in., on high plateaus near Diamond Butte on the Custer National Forest), a difference of about 50% compared to the lower value. Precipitation patterns for the Baker, Bridger and Ekalaka stations (Figure 1) indicate that the amount received in the biologically critical spring quarter (April, May, June) ranges from 44 to 47% of total precipitation. These percentages are almost identical to the spring percentage (and absolute amount) received in southwestern Montana, where sagebrush also predominates (Lesica et al. 2005). Due to the distance from moderating oceanic influences, another semi-arid climatic regime attribute is strong seasonal (winter to summer) and diurnal temperature fluctuations. The main climatic difference between Wyoming big sagebrush habitats in eastern Montana and in southwestern Montana is the warmer summer daily maxima and minima in eastern Montana, due primarily to lower elevations. Eastern Montana study area elevations ranged 270 to 1,220 m (890 to 3,990 ft.). Sampled plots in Southwestern Montana sagebrush ranged from 1,800 to 2,035 m (5,900 to 6,675 ft.; Lesica et al. 2005). Both regions reliably experience convectional storms in July and August, but rainfall is locally erratic within both areas.

Wyoming big sagebrush was the only big sagebrush subspecies identified on sampling sites, although the considerably larger silver sagebrush was also encountered, especially on stream terraces and sites having a greater percentage of sand in the soil. In addition to being distinguished by minor morphologic and chemotaxonomic differences from silver sagebrush, Wyoming big sagebrush also occurs on more xeric sites where the annual precipitation ranges from 18 to 30 cm (7-11 in.) (Winward 2004). Modeled study area annual precipitation ranges from 274 mm (10.8 in.) to 415 mm (16.35 in.) (Thornton et al. 1997). This apparent range extension of Wyoming big sagebrush in terms of precipitation values may be explained by its occupying higher elevation sites in southeastern Montana, a region beyond the established geographic range of mountain big sagebrush. These extremes in precipitation combine with other site differences, such as elevation (an indirect measure of precipitation and evapotranspiration), slope, aspect, and soil texture (as measure of available water capacity) to explain the range in mature plant height, 35 to 105 cm, and the diversity of plant associations noted across sampling sites.

The plant association associated with the driest sites (mostly due to their very well drained soils) was Wyoming big sagebrush / bluebunch wheatgrass (*Pseudoroegneria spicata*). The most mesic sites are characterized by Wyoming big sagebrush / Idaho fescue (*Festuca idahoensis*) – western wheatgrass (*Pascopyrum smithii*); these sites were found on the relatively high elevation butte tops of the Custer National Forest where average annual precipitation exceeds 400 mm (16 in.). The most commonly encountered plant

Figure 1. Map of study area.



association was Wyoming big sagebrush / western wheatgrass – green needlegrass (*Nassella viridula*). Other plant associations are permutations of this type created by site conditions (predominantly related to soil texture) and disturbance regimes. These include Wyoming big sagebrush / western wheatgrass – Sandberg's bluegrass (*Poa secunda*), Wyoming big sagebrush / western wheatgrass – blue grama (*Bouteloua gracilis*), Wyoming big sagebrush / western wheatgrass – needle-and-thread (*Hesperostipa comata*), and Wyoming big sagebrush / western wheatgrass.

Methods

Field Methods

In June and July of 2006 and 2007 we sampled 24 sites dominated by Wyoming big sagebrush in central and southeastern Montana within Big Horn, Carbon, Carter, Custer, Garfield, McCone, Petroleum, Phillips, Powder River, Rosebud and Yellowstone Counties. We used lists of potential sites provided by the Miles City Office of the Bureau of Land Management (BLM), the Ashland Ranger District of the Custer National Forest, and personal communication to select sampling sites based on age of burn and accessibility. Larry Eichhorn, retired BLM range conservationist from Lewistown, provided information valuable in relocating the original sample sites of his study of post-fire succession in central Montana (Eichhorn & Watts 1984). We focused on federally or state owned lands but did find several cooperative private landowners.

At each site a macroplot (20 m by 50 m, 1000 m^2) was visually selected to represent prevailing conditions within the burned area. A control sample macroplot was established in unburned sagebrush-dominated vegetation as close as possible to the burn. The control was chosen to be as similar as possible to the abiotic setting (slope, aspect, soils) of the burned sample plot. Although the unburned control macroplots are not true controls because of not being randomly assigned prior the fires, nonetheless they function as controls by exemplifying what the burned plot probably would constitute, had they not burned. With one exception, burned macroplots were located within 20 m or less of the unburned control and always in the same grazing pasture (not separated by fencing). We noted the positions of fence lines and water developments and attempted to locate sampling points as far removed as possible to ensure that grazing pressure was not excessive. However, we had no way of accurately accounting for grazing regimes.

We used the Daubenmire (1959) concept of canopy cover to estimate this parameter along five evenly-spaced, parallel 20 m transects originating at the 50 m macroplot baseline (Mueller-Dombois and Ellenberg 1974). At the 5 and 10 m marks of the five transect lines 3 m^2 circular microplots were established for determining rooted density for all shrub species by four size classes: 1) seedlings, height < 10 cm; 2) juveniles, height > 10 cm and stem diameter at ground level < 1 cm; 3) sub-adults, stem 1-3 cm diameter and 4) adults, > 3 cm stem diameter. At alternate microplots (total of five microplots) age and height were recorded for one sagebrush plant of each size class; we focused on specimens exhibiting the least crown damage. Sagebrush plants were cut with a fine-blade saw or sharp pruning shears at ground level (which sometimes required removing accumulated detritus from around mature stems). Annual growth rings were field counted with a 10X or 20X hand lens (Ferguson 1964). To ensure that we had at least three estimates for each size class it was necessary to sample sagebrush plants outside the microplots, however, complications to this approach arose due to a tendency for even-aged stands and damaged or rotted stems where we could not reasonably approximate age. Study area Wyoming big sagebrush plants tended toward a deliquescent form (especially sub-adult and adult classes). This tendency combined with mechanical damage from grazers (presumably domestic stock) results in stems lacking the pith and some number of annual rings. We frequently experienced stands where all sub-adult and adult specimens, or at least the 10 to 20 specimens we cut, were incapable of being accurately aged and the smaller size classes had significant stem damage as well.

In these same ten microplots we estimated the percent canopy cover (Daubenmire 1959) of all

vascular plant species and ground cover types (bare soil, gravel, rocks, litter, lichens, mosses, basal vegetation) using 13 cover classes (T- = >0, ≤0.1; T = >0.1, ≤1; P = >1, ≤5; 1 = >5, ≤15; 2 = >15, ≤25, 3 = >25, ≤35; 4 = >35, ≤45; 5 = >45, ≤55; 6 = >55, ≤65; 7 = >65, ≤75; 8 = >75, ≤85; 9 = >85, ≤95; F = >95, ≤100%). Also recorded was the number of occurrences of each species of the Cichorieae tribe of the Asteraceae in all ten microplots.

The only burn information recorded was ignition source, either wildfire or prescribed burn, and the year of occurrence. Attempts to characterize fire severity, a potentially significant explanatory variable, were difficult because 1) immediate post-fire conditions were not generally recorded, and 2) quite a number of the burn ages were old enough (20+ years) that significant clues had been obscured. Fires presumed to be of high-intensity consumed all, or nearly all of the sagebrush stems, leaving only 2-5 cm projecting above the ground and, in the most extreme cases, created a concave stem obscured by surface materials. For two sites the only evidence of fire was very scattered charred branch remains and an obvious fire-line; no stumps could be located. Both wildfires and prescribed fires resulted in fire effects categorized as high-severity. Fires of presumed lesser intensity resulted in standing sagebrush main stems with secondary and tertiary branches intact, but charred. Examination of larger burns commonly revealed multiple burn severity levels (so far as we were able to detect these effects given the long time since burning). Several sampling sites contained a few Wyoming big sagebrush specimens that gave the appearance of having escaped burning; only one specimen was reliably aged and removed from the recovery figures, the rest were counted as part of the recovered cohort when sampling procedures encountered them (aging indicated they had established post-burn).

Data Analysis

Our main emphasis was to describe Wyoming big sagebrush recovery, which we characterize as percentage recovery and is calculated by using the mean canopy cover or height of this subspecies for the burned macroplot divided by values from the unburned control macroplot. We evaluated changes in stand height by using the size class with the greatest canopy cover. Rate of recovery for sagebrush is calculated as the percent recovery for either canopy cover or height divided by the number of years since burning. A planned demographic analysis was frustrated by our inability to accurately age stems, except those of the seedling size class. Species richness is measured by the number of vascular plant species recorded in the 5 line intercepts (shrubs only) and ten microplots (total of 30 m2).

The relative aridity of a site, as measured by precipitation and potential evapotranspiration, was hypothesized to affect recruitment and other aspects of stand recovery. Slope and aspect are the primary determinants of potential evapotranspiration; these two variables along with latitude have been integrated into a "heat loading" index by McCune and Keon (2002). Average site annual precipitation was estimated by DAYMET a statistical model that integrates elevation, other aspects of local terrain, and geographic position with weather station data for the past 20 years (Thornton et al. 1997).

We used paired-sample t-tests to evaluate the differences between burned and unburned control macroplots for Wyoming big sagebrush canopy cover and height, total shrub cover and cover of perennial grasses, annual grasses and forbs. Linear regression analysis was used to model the recovery of sagebrush height, sagebrush canopy cover and herbaceous cover with time since fire. When modeling sagebrush recovery regression lines were forced through the origin to reflect biological realities. Regression analysis was also used to test the association between recovery rate of sagebrush and the abiotic site factors of precipitation, heat load index and soil texture.

Results

Wyoming Big Sagebrush and Shrub Recovery

The sampled sites span a wide range of sites in terms of water stress and hence composition. A

50% difference in annual precipitation, 10.8 to 16.3 cm, across the range of sites is probably the major driver of compositional and canopy cover values. The comparatively minor difference in heat load index, 12%, between the most "extreme" sites in our dataset is to be expected in these rolling plains where the steepest slope was only 11%. Assuming the kind and amount of undergrowth vegetation is indicative, then the driest sites dominated by bluebunch wheatgrass (averaging 28% cover of perennial grasses with a range of 22 to 35%, Appendix B) can be contrasted with the high-elevation sites dominated by Idaho fescue and western wheatgrass (74% average canopy cover perennial grasses, ranging from 68 to 80%). The remainder of the control plots did not always fall between these extremes of perennial grass cover, probably because of grazing effects (both sampling year and long-term). Values lower than those listed for bluebunch wheatgrass-dominated control plots were registered for a number of control plots having western wheatgrass dominant in several different plant associations.

There was a median time since fire of 22 ± 16 (16 = 1 std. dev.) years, ranging from 4 to 67 years for the 24 paired macroplots (control and burned) we sampled. Fire resulted in a virtually complete loss of shrub canopy cover as revealed by examination of recently burned macroplots (≤10 years, N = 6); five of the six plots had no shrub canopy cover and one had < 2%. Wyoming big sagebrush is the dominant shrub on the control macroplots with an average cover of 20 ± 8%; total shrub cover averages only slightly more, 21 ± 8%, with the additional species including silver sage, rubber rabbitbrush (*Ericameria nauseousa*), green rabbitbrush (*Chrysothamnus viscidiflorus*), and spineless horsebrush (*Tetradymia canescens*). Silver sage is the only shrub even approaching Wyoming big sagebrush in cover and that occurred on only one site. The average height of the dominant cohort of Wyoming big sagebrush in control plots was 61±11 cm.

Because there were only five prescribed burn sites and four of these showed no recovery in Wyoming big sagebrush cover (or total shrub cover) we did not stratify the dataset according to mode of ignition. For the recovery of Wyoming big sagebrush canopy cover a linear model (Figure 2) resulted in the best fit with age since fire explaining 29% of the variation in cover (t = 2.81, P = 0.010). For total shrub cover recovery results were not much different with a linear model explaining only 22% of the variation (t = 2.38, P = 0.027). The mean recovery rate for Wyoming big sagebrush canopy cover was 0.16% / year ± 0.45; projecting this rate results in a predicted 100%

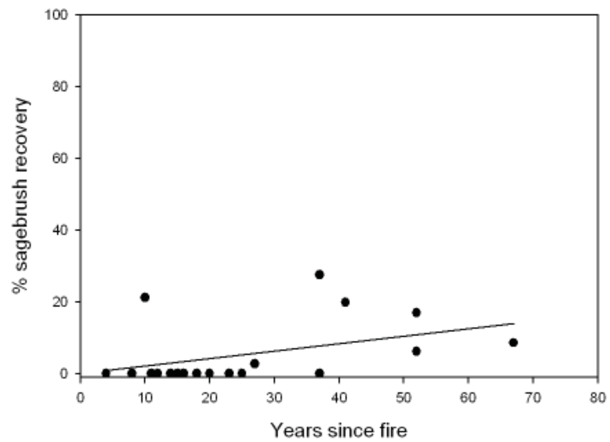


Figure 2. Linear model between Wyoming big sage percent canopy recovery and time since fire for 24 sites (both prescribed and wildfire); regression model constrained to pass through the origin.

recovery requiring an average of 625 years. Height recovery of the dominant Wyoming big sagebrush cohort was best fit by a highly significant (t = 4.81, P = <0.001) second order function (Figure 3) in which time since fire explained 55% of the variation and the extrapolated time of recovery is approximately 68 years. We also present a linear model for comparison (Figure 4) that explains 54% of the variation and is highly significant (t = 4.83, P = <0.001), but which yields an intercept of more than 80 years for complete height recovery, a result inconsistent with biological realities.

A linear regression model incorporating the heat load index and mean annual site precipitation explained 30% of the variation in the rate of Wyoming big sagebrush canopy recovery. However, neither annual precipitation (P = 0.827) nor the heat index load (P = 0.54) alone were significantly related to canopy recovery rate.

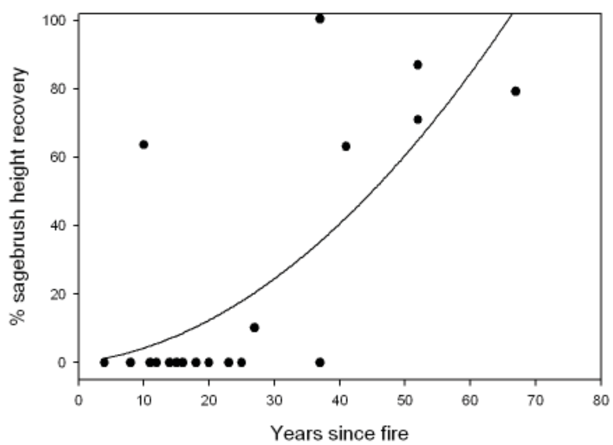


Figure 3. A second order function depicting canopy height of Wyoming big sage dominant cohort since fire for 24 sites; regression model constrained to pass through the origin.

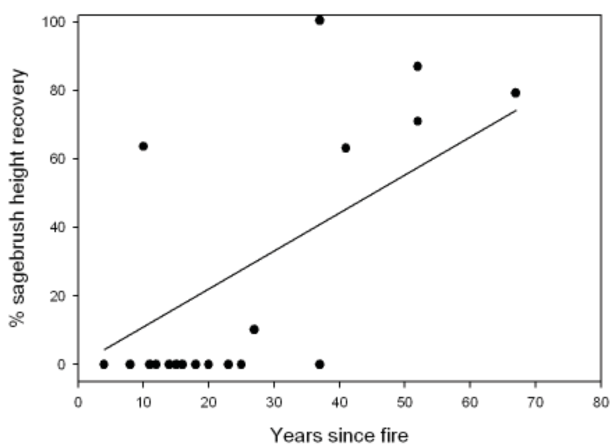


Figure 4. Linear model between canopy height of dominant Wyoming big sage cohort and time since fire for 24 sites (prescribed and wildfire); regression model constrained to pass through the origin.

Although we were unable to acquire accurate ages, regardless of specimen maturity/size class, Table 1 presents our best estimates of age as well as density and height by maturity class. The density (# of stems / m^2) of Wyoming big sagebrush is highly variable across all size classes in both burn and control macroplots. The average density of burned macroplots as a percentage of control plots ranges from 25% for seedlings to 1% for the adult class. The adult class dominates the structure for the control plots but the variation in structure is considerable. For burned macroplots no one class is dominant. Obscured in Table 1 by the averaging process is the fact that of the 24 burned macroplots the seedling class was represented in only 4 macroplots, the juvenile class in 5 macroplots, the subadult class in 6 macroplots, and the adult class in 4 macroplots.

Herbaceous Recovery

The important perennial graminoids in order of declining constancy were western wheatgrass, Sandberg's bluegrass, blue grama, prairie junegrass (*Koeleria macrantha*), green needlegrass, needle-and-thread, bluebunch wheatgrass and sun sedge (*Carex inops* ssp. *heliophila*). The mean perennial grass canopy cover on control macroplots was 40%, approximately half was western wheatgrass. The burned macroplots had an average of 61% perennial grass cover, 39% of that cover is western wheatgrass. The difference in perennial grass cover was highly significant (t = 4.83, P <0.001), but the time since fire is insignificant (t = 1.29, P = 0.179) in explaining the difference.

The annual grass component, with an average cover of 19% and 9% in burned and control macroplots respectively, is comprised primarily of the introduced brome grasses (field brome, formerly Japanese brome) and cheatgrass (*Bromus tectorum*). However, the native sixweeks fescue (*Vulpia octoflora*, formerly *Festuca octoflora*) also has appreciable constancy, although its cover is negligible. The difference in annual grass

Table 1. Demographic parameters for Wyoming big sage on burned and control plots; averages and ranges by four size/maturity classes.

	Size/maturity classes											
	Seedling			Juvenile			Sub-adult			Adult		
Burn versus Control Plots	Number/ m2 (range)	Height (cm)	Average Age (Range)	Number/ m2 (range)	Height (cm)	Average Age (Range)	Number/ m2 (range)	Height (cm)	Average Age (Range)	Number/ m2 (range)	Height (cm)	Average Age (Range)
Burn plots	0.03	6 ± 2	2.6	0.01	19 ± 8	7.0	0.03	33 ± 6	18.6	0.01	51 ± 7	27.5
	(0 to 0.27)		(2 to 3)	(0 to 0.06)		(4 to 9)	(0 to 0.13)		(12 to 25)	(0 to 0.06)		(28 to 35)
Control Plots	0.12	5 ± 2	6.1	0.24	17 ± 3	10.0	0.52	36 ± 8	19.3	0.64	62 ± 12	35.9
	(0 to 0.63)		(2 to 13)	(0 to 0.83)		(4 to 18)	(0 to 1.43)		(8 to 45)	(0.23 to 1.27)		(18 to 71)

cover between burned and control macroplots is significant (t = 2.818, P = 0.010), but the difference cannot be significantly attributed to time since burning (t = 1.038, P = 0.311).

For forbs, there was no statistical difference (t = 0.132, P = 0.896) in average canopy cover between burned (8.3%) and control (8.0%) macroplots. Forb canopy cover was less than graminoid cover, but it did range as high as 27% due to an unusual post-fire increase in the non-native corn speedwell (*Veronica arvensis*) on one productive high-elevation site. The most common forbs are the non-natives pale madwort (*Alyssum alyssoides*), field cottonrose (*Logfia arvensis*, formerly *Filago arvensis*), herb sophia (*Descurainia sophia*), littlepod false flax (*Camelina microcarpa*), yellow salsify (*Tragopogon dubius*), common dandelion (*Taraxacum officinale*), and the natives woolly plantain (*Plantago patagonica*), tiny trumpet (*Collomia linearis*), rough false pennyroyal (*Hedeoma hispida*), spiny phlox (*Phlox hoodii*) and American vetch (*Vicia americana*).

The most commonly occurring forbs of the Cichorieae tribe were the non-natives common dandelion and yellow salsify. The natives weevil prairie-dandelion (*Nothocalais troximoides*) and pale agoseris (*Agoseris glauca*) were found in only one and two plot pairs, respectively. Due to their extremely low densities native species were lumped with non-natives for analysis. The mean density of members of the Cichorieae tribe was 1.8 ±2.8 plants / m^2 for the burned macroplots and 1.3 ± 1.8 plants / m^2 for the unburned controls; this difference was not significant (N = 24, t = 0.448, P = 0.659).

Species Richness

The average number of species per macroplot, species richness, had a mean value of 32 ± 6 for control plots and 26 ± 7 for burned plots. Extreme values ranged from 19 to 44 and 12 to 40 for control and burned plots, respectively. There was a significant difference in species richness between burned and control macroplots (t = 3.737, P = 0.001), however this difference was not associated with time since burning (t = 0.588, P =0.563).

Discussion

Sagebrush and Shrubs

Observation of both recently burned stands and those of considerable post-burn age (> 20 years) indicate that Wyoming big sagebrush mortality was virtually complete. There was no measured canopy recovery for Wyoming big sagebrush in 17 of the 24 sites. Our linear model of canopy recovery is based on 24 sample pairs and the indicated recovery rate is exceedingly slow. The highest recovery rate in our study, 0.72 % / year (27% recovery in 37 years), still implies full recovery would require much more than 100 years given the linear model. The oldest burn, 67 years, was only 8% recovered and recovery on the most moisture-stressed sites as well as sites with the greatest precipitation and most mesophytic vegetation composition registered no recovery within 14 years. Even on an older (27 years), and ostensibly cooler prescribed burn, recovery was only 3%. The only site (Little Bighorn Battlefield National Monument) without domestic stock use within the recovery period (and for a considerable period prior to burning) recorded no shrub canopy recovery in 25 years.

In the only other study within our sampling area, Eichhorn and Watts (1984) found no re-establishment of Wyoming big sagebrush in the 14 years following wildfire in the Missouri River Breaks and vicinity. In southwestern Montana, Wambolt et al. (2001) reported a 72% recovery of Wyoming big sagebrush after 32 years in one burn and 96% recovery after only nine years in another. Watt and Wambolt (1996) documented 76% recovery within 30 years in another southwestern Montana study. It should be noted these southwestern Montana studies documented cool-season, prescribed fires. Also in southwestern Montana, Lesica et al. (2005) documented almost no Wyoming Big Sagebrush canopy recovery in six wildfire burn plots, the most being 3% in 23 years. In southeastern Idaho, Colket (2003) found, measuring density not cover, that 3 of 17 plots attained full recovery in 53 years and that by 92 years 16 of the 17 plots reached full density. Attaining full density is not equivalent to recovery

of canopy cover, which undoubtedly would require additional decades for shrubs to mature (Baker 2007).

A nearby seed source is generally regarded as promoting faster stand recovery (Blaisdell 1953, Gruell 1980) because the seed bank of *A. tridentata* is negligible to non-existent (Young and Evans 1989, Akinsoji 1988). For these reasons we located the burn sample plots as close as practicable to control plots, the ostensible seed source. This strategy apparently made no difference, similar to the results of Wambolt and Payne (1986) in their prescribed burn study where the close proximity of seed source still resulted in no Wyoming big sagebrush re-establishment six years post-burn.

We hypothesized that stands on areas of higher precipitation and/or with a lower heat load index would have a higher rate of recovery, similar to results from Johnson and Payne (1968). However, we were unable to detect any biotic or abiotic variables associated with Wyoming big sagebrush recovery across our study area. A model with age since fire, heat load index and precipitation explained 30% of the variation in canopy recovery, however, almost all of this explained variation was attributable to using age as a covariate.

The average height of the dominant Wyoming big sagebrush cohort in control plots is 61 ± 11 cm, which agrees well with our southwestern Montana (Lesica et al. 2005) measures of this subspecies (61 ± 6 cm). Only 4 of 24 burned macroplots even had a mature size/age class represented and the average height was 50 ± 6 cm; one burned macroplot attained full height recovery in 38 years. Removing the zero values for height recovery from Figure 3 would obviously shorten the time expected for full recovery and would more closely model what would be expected in the rate of height growth of individual plants once established on a site. However, for the model to be realistic on a stand basis the zero values should be included.

The results of the demographic portion of this study are disappointing due to equivocal aging of the sagebrush. The poor condition (loss of innermost annual rings, misshapen crowns) of sagebrush stems spanned all age/size classes, but defects were especially pronounced in the adult class. More than 80% of burned macroplots lack any representation of a seedling class. Seedling production was virtually nil, even in 2007, a year with abundant spring moisture that should have favored at least seedling germination, if not survival. We questioned this lack of seedlings as perhaps anomalous and a consequence of inadequate sampling. Therefore, in addition to visually examining the 10 microplots, we conducted extensive searches of adjacent terrain and uniformly failed to detect seedlings there as well. In general, the control macroplots had all maturity classes represented, however more than 50% of the stands did not have a seedling class present. The considerable difficulty Wyoming big sagebrush exhibits in site recolonization might be expected given that it occupies the driest sites with the most poorly developed soils (Morris et al. 1976, Barker and McKell 1983).

With the exception of the mostly missing seedling class, nearly all the unburned control plots were uneven-aged (had multiple size/maturity classes represented), revealing recruitment is not limited to immediate post-fire circumstances. Three control plots were somewhat anomalous in that only an adult class was present. Two of these three plots had approximate stem ages indicating that there had been no recruitment in more than 25 and 40 years. Although the adult class of these two stands was not even-aged, their age structure suggests episodic reproduction at some point in time. Two of these three stands were noted to have considerable Wyoming big sagebrush mortality of undetermined cause (visually perceived to be greater than noted for other sample stands).

Graminoids

The highly significant 21% increase in perennial grass cover shows no diminution with time since fire, which is understandable given that Wyoming big sagebrush cover exhibits hardly any recovery, even after more than 60 years. This response can be contrasted with perennial grass cover in burned stands once dominated by mountain big sagebrush in southwestern Montana where there was a modest 7% increase in perennial grasses (Lesica et al.

2005). However, this effect was not detectable after about 25 years, about a decade preceding full sagebrush canopy recovery (Lesica et al. 2005). Possibly even greater increases in annual grass cover may have been negated by post-fire livestock grazing when the grass becomes more accessible after shrub canopy elimination (Pechanec et al. 1954, Harniss and Murray 1973, Bunting et al. 1998). In our unburned control plots perennial grass cover, an index of long-term grazing intensity, was not associated with proportional changes in grass cover following fire. This implies that post-fire grazing has not had a large impact on fire-induced changes.

The major contributor to the significant increase in post-fire perennial grass cover appears to be the rhizomatous western wheatgrass with a highly significant (N = 23, P = 0.001) 17% difference (39 vs 22%) in cover (77% increase). The other important rhizomatous graminoid, blue grama, exhibited 10% average canopy cover on burn plots and only 4% on controls, but due to high variability this difference is not significant at the 5% level (N = 22, P = 0.085).

The species richness of bunch-forming graminoids is greater than that of the rhizomatous component, although their combined canopy cover is less in burned (12%) and control (14%) macroplots. Sandberg's bluegrass is highly constant but insignificant in cover, both in burned (1.4%) and control (1.3%) macroplots, and shows no significant response to burning (N = 24, P = 0.752). In several stands green needlegrass registered a large post-fire cover increase, but overall there was no significant effect (N = 18, P = 0.155). The 5% average canopy cover of needle-and-thread in both control and burned macroplots reflects no significant difference (N = 16, P = 0.915), but there were both notable six-fold increases and a fifty-fold decrease. Bluebunch wheatgrass, a relatively less important grass in the study area (45% constancy), gives the deceptive impression of decreasing canopy cover with burning (2% vs. 5%), but in at least one instance cover notably increased, resulting in overall statistical insignificance (P = 0.345). This inconsistent bluebunch wheatgrass response with burning reflects results found in the literature with increases (Wambolt and Payne 1986, Humphrey 1984), decreases (West and Hassan 1985) and no change (Peek et al. 1979, Antos et al. 1983) recorded. The lumping of prescribed with wildfire responses in our test may have resulted in a seeming lack of bluebunch wheatgrass cover association with fire. For example, to the west of the Musselshell River's confluence with Ft. Peck Reservoir on the 1996 Alkali Creek Burn, a wildfire of presumed high-intensity nearly extirpated bluebunch wheatgrass (decreasing to 0.25% from the 29% canopy cover on the control). Emphasizing the uniqueness of fire response is the observation that on the same plot pair blue grama cover increased dramatically (control 2%; burned 52%), presumably as a result of fire.

Idaho fescue is an important grass in eastern Montana only on high-elevation sites. On our two sites, both with prescribed fires, it both increased dramatically (47 to 69%) and decreased (38 to 23%) 14 and 15 years post-fire, respectively. In southwestern Montana there was no statistical difference in Idaho fescue cover between burned and control macroplots (Lesica et al. 2005), although it has been reported that this species is damaged by fire, at least in the short-term, due to the foliar density of tussocks (Wright et al. 1979).

The average annual grass canopy cover for both burned (19%) and control (9%) macroplots is comprised almost entirely of the non-native field or Japanese brome, which has a highly significant cover increase following fire (N = 23, P = 0.010), and no significant diminution of cover with time since fire. Its cover ranges from zero to 69% in burned plots. Field brome is usually regarded as a weed on rangelands and prairies because it competes with native perennials for water and nutrients (Stubbendieck et al. 1985, Gartner et al. 1976). Fire is noted (Gartner et al. 1986, Whisenant 1990) to reduce field brome population density for one or two years post-burn primarily as a consequence of litter reduction (critical for seed germination and establishment). We found no research that followed the post-burn course of succession for more than two years. We hypothesize that the observed field brome response was due to exploitation of space, water

and nutrients following sagebrush mortality and consequent loss of competition.

Forbs

Our results suggest that forbs are generally well-adapted to these fire-prone communities because no statistical difference was demonstrated ($t = .132$, $P = 0.896$) between burn (8%) and control (8%) macroplot forb cover. However, we did have plots where forb cover decreased or increased drastically, usually due to the cover of one or two species. For example, on both the youngest (4 years) and oldest (67 years) burns lesser spikemoss (*Selaginella densa*) was totally killed and reduced to 2% cover contrasted with 41% and 22% cover, respectively, on the control macroplots. On two plot pairs a positive response to burning was displayed by the annual non-natives field cottonrose (<1% to 24%) and corn speedwell (<1% to 23%). The rather stochastic nature of these responses is emphasized by the fact that field cottonrose cover was minor (1.5%) in the burned macroplot where corn speedwell cover was so high. It is noteworthy that these large differences in forb cover are due to annuals, not to native perennials, which register hardly any change. Similar results have been reported for prescribed burns in sagebrush steppe by Peek et al. (1979), who found forb frequency was not affected three years post-burn, and also by the Harniss and Murray (1973) report of stable forb cover for 30 years following fire in eastern Idaho. Wildfire did not produce any change in canopy cover of forbs in south-central or southwest Montana (Hoffman 1996, Fraas et al. 1992).

Forbs of the Cichorieae Tribe of the Asteraceae Family have been determined to comprise an important component of Greater Sage-Grouse summer diet and are often crucial for successful brood rearing (Klebenow and Gray 1967, Peterson 1970, Barnett and Crawford 1944, Drut et al. 1994). An increase in forbs can be expected with the fire-induced reduction in the cover of shrubs and grasses (Klebenow 1973, Glenn-Lewin et al. 1990). We combined the relatively rare occurrences (<40 plants / 1,440 m^2) of native Cichorieae weevil prairie-dandelion and pale agoseris with the much more abundant non-native Cichorieae densities, but found no evidence for a fire-driven change. Lesica et al. (2005) also found no change with fire for non-native Cichorieae in southwest Montana (2.7 ± 0.9 plants / m^2, burned macroplots: 2.0 ± 0.6 plants / m2, control plots). Comparable figures for our study area are 1.6 ± 2.7 plants / m^2 (burned) and 1.3 ± 1.8 plants / m^2 (control), which indicates that study area Cichorieae densities are less than those of southwestern Montana and considerably more variable site to site. A high degree of within site variation in density was also noted, but not statistically tested. Since non-native Cichorieae are invasive and increase with disturbance (Hobbs and Huenneke 1992, Kotanen et al. 1998) their lack of response was unexpected.

Management Implications

Most research from outside our study area, documents a highly variable response of Wyoming big sagebrush to prescribed burning and a variable response of longer recovery periods for wildfire. Our data from central and southeastern Montana suggest that recovery (attaining 100% canopy cover of control) will require much more than 100 years. We had no rapid Wyoming big sagebrush recovery within the study area. The only other data within our study area (Eichhorn and Watts 1984) indicated no Wyoming big sagebrush recovery in 14 years and is corroborated by our study showing that even our oldest prescribed burn, which also occurred on a mesic site, had only 3% recovery in 27 years. The response to wildfire may be even slower with two of our sites showing no recovery 23 and 25 years following burning, and our oldest sites had only 6 to 17% recovery after > 50 years. The average Wyoming big sagebrush canopy recovery rate of $0.16 \pm 0.44\%$ / year implies full recovery is attained in > 600 years which is biologically improbable because as demonstrated by Lesica et al. (2005, who had data-points from the complete time-line including full recovery) a non-linear model is the best fit. Although Lesica et al. (2005) documented mountain big sagebrush recovery, we presume the model expression would be similar for Wyoming big sagebrush with only a greater time to full recovery, certainly less than 600 years. In interpreting our results it should be noted that our close placement (<20 m) of burn

macroplots to control plots should speed recovery due to local seed source proximity. An example of differential recovery with distance from seed source is detailed by Welch and Criddle (2003). Mountain big sagebrush canopy recovery takes about 35 years, but to merely reach the interior of a burn in Idaho required 70 years or more (Welch and Criddle 2003). The time to fully recover an extensive Wyoming big sagebrush burn could be very considerable.

The three stands with only an adult size class present might be considered as evidence supporting the contention that sagebrush steppe is a fire-dependent vegetation type requiring periodic renewal by fire (Winward 1991). Overall the size class structure of our stands argues for a steady-state structure and a lack of fire dependence as suggested by Connelly et al. (2000) and Welch and Criddle (2003). Our results support the observation that, although fire is an important natural disturbance in sagebrush steppe, it could not have occurred as often as suggested in the past (see Baker 2007 for a review). Our results support Baker's (2007) interpretation indicating that fire rotations are about 100 – 240 years for Wyoming big sagebrush and that sagebrush steppe belongs to fire regime V (long rotation, stand replacement).

None of the factors (soil texture, precipitation, slope, aspect [we combined slope and aspect into a heat load index]) that have been cited as influencing sagebrush recovery (Johnson and Payne 1968, Gruell 1980) were associated with the rate of canopy recovery in our study. Thus, managers cannot presume that stands of Wyoming big sagebrush on more mesic sites will exhibit faster recovery, or that prescribed fire, as compared to wildfire, will result in more rapid recovery.

Our results are pertinent to protecting native biological diversity and managing domestic stock within the study area sagebrush steppe. An average increase in perennial grass cover of 27% and 20% followed prescribed fire and wildfire, respectively. We have no evidence that this amplified cover will be diminished until sagebrush canopy cover becomes substantial at some future time, probably at least a century after burning. Greater Sage-Grouse will find this augmented perennial grass cover beneficial (Wallestad and Pyrah 1974, Aldridge and Brigham 2002) as will domestic stock, which also benefit from increased accessibility to the herbaceous component due to shrub canopy removal. The 11% increase in annual grasses is due almost wholly to field brome which is considered by some a noxious weed (Stubbendieck et al. 1985) because it competes with native perennials for water and nutrients and has a brief window of grazing availability as it rapidly matures and loses nutrient content, digestibility and palatability (Stubbendieck et al. 1985). Although various studies (see Stubbendieck et al. 1985) indicate it declines with time on a site we have no indication this is the case. Burning sagebrush stands infested with field brome may result in a long-term increase in this undesirable species.

Success of Greater Sage-Grouse brood rearing is dependent on available forbs, especially those of the Cichorieae, both native and exotic (Connelly et al. 2000). We found no predictable increase in forb cover, including those of the Cichorieae, with fire. At some sites we did find a large increase in exotic annual forbs, presumably they consume water and nutrients better directed to perennial natives and they appear unpalatable to domestic stock as well.

Managers concerned about declining populations of Greater Sage-Grouse and some other sage-dependent species should be aware of the Wyoming big sagebrush response after fire in our study area. Greater Sage-Grouse are dependent on some mixture of open- and closed-canopy sagebrush habitats to complete their life cycle (Connelly et al. 2000). Wyoming big sagebrush recovery takes so long that managers considering prescriptive burns need to have a long-term view of the landscape before eliminating a sagebrush habitat that will not return for at least a century. Similar concerns may be expressed about wildfire management in sagebrush habitats.

Conclusion

Wyoming big sagebrush recovery from prescribed fire and wildfire was extremely slow in our

eastern Montana study area and likely requires well over 100 years to reach pre-burn sagebrush cover conditions. Results were similar across all environmental conditions, even at relatively mesic sites. Perennial and annual grass cover increased after burning, but the annual grass increase consisted almost entirely of field (Japanese) brome, a non-native that is considered a weed with negative habitat and livestock value. Forbs, most especially those of the Cichorieae tribe of the Asteraceae family, are important for Greater Sage-Grouse brood rearing; however, we found no predictable change of this component with fire. Plant species richness was lower in burned plots. Resource managers concerned about Greater Sage-Grouse and other sage-dependent species should carefully consider the long-term ramifications of prescribed burns and the effect of wildfires on Wyoming big sagebrush habitat in eastern Montana. Burns may essentially eliminate sagebrush habitat, increase weedy annual grass cover, reduce species richness, and could require a century or more for recovery to pre-burn sagebrush cover conditions.

Literature Cited

Akinsoji, A. 1988. Postfire vegetation dynamics in a sagebrush steppe in southeastern Idaho, USA. Vegetatio 78: 151-155.

Aldridge, C. L. and R. M. Brigham. 2002. Sage grouse nesting and brood habitat use in southern Canada. Journal of Wildlife Management 66: 433-444.

Antos, J. A., McCune, B., and C. Bara. 1983. The Effect of Fire on an Ungrazed Western Montana Grassland. American Midland Naturalist 110: 354-364.

Bailey, R. G. 1995. Descriptions of the ecoregions of the United States. Second Edition. U. S. Department of Agriculture, Forest Service. Misc. Publ. 1391.

Baker, W. L. 2007. Fire and restoration of sagebrush ecosystems. Wildlife Society Bulletin. 34(1): 177-185.

Barker, J. R. and C. M. McKell. 1983. Habitat differences between basin and Wyoming big sagebrush in contiguous populations. Journal of Range Management 36: 450-454.

Barnett, J. K. and J. A. Crawford. 1994. Pre-laying nutrition of sage grouse hens in Oregon. Journal of Range Management 47: 114-118.

Blaisdell, J. P. 1953. Ecological effects of planned burning of sagebrush-grass range on the upper Snake River Plains. USDA Technical Bulletin 1975. Washington, DC.

Bunting S. C., Robberecht R. and G. E. Defossé. 1998. Length and timing of grazing on postburn productivity of two bunchgrasses in an Idaho experimental range. Int. J. Wildland Fire 8, 15-20.

Colket, E. C. 2003. Long-term vegetation dynamics and post-fire establishment patterns of sagebrush steppe. Thesis, University of Idaho, Moscow.

Connelly, J. W., M. A. Schroeder, A. R. Sands and C. E. Braun. 2000. Guidelines to manage sage grouse populations and habitat. Wildlife Society Bulletin 28: 967-985.

Daubenmire, R. 1959. A canopy-coverage method of vegetational analysis. Northwest Science 33: 43-64.

Drut, M. S., W. H. Pyle and J. A. Crawford. 1994. Diets and food selection of sage grouse chicks in Oregon. Journal of Range Management 47: 90-93.

Eichhorn, L. C. and C. R. Watts. 1984. Plant succession on burns in the river breaks of central Montana. Proceedings of the Montana Academy of Science 43: 21-34.

Eng, R. L. and P. Schladweiller. 1972. Sage grouse winter movements and habitat use in central Montana. Journal of Wildlife Management 36: 141-146.

Ferguson, C. W. 1964. Annual rings in big sagebrush. University of Arizona Press, Tucson.

Fraas, W. W., C. L. Wambolt and M. R. Frisina. 1992. Prescribed fire effects on a bitterbrush-mountain big sagebrush-bluebunch wheatgrass community. Pages 212-216 in: W. P. Clary et al., compilers. Proceedings--symposium on ecology and management of riparian shrub communities. USDA General Technical Report INT-289. Ogden, Utah.

Gartner, F. R., White, E. M., and R. L. Butterfield. 1986. Mechanical treatment and burning for high quality range forage. Beef Report: Cattle 86-29. Brookings, SD: South Dakota State University, Department of Animal and Range Sciences and Plant Science; Agriculture Experiment Station: 135-140.

Gartner, F. R., Roath, L. R., and E. M.White. 1976. Advantages and disadvantages of prescribed burning. *In*: Use of prescribed burning in western woodland and range ecosystems: Proceedings of a symposium; 1976; Logan, UT. Logan, UT: Utah State University: 11-15.

Glenn-Lewin, D. C., L. A. Johnson, T. W. Jurik, A. Kosek, M. Leoscheke, and T. Rosburg. 1990. Fire in central North American grasslands: vegetative reproduction, seed germination and seedling establishment. Pages 28-45 *in* S. L. Collins and L. L. Wallace editors, Fire in central North American grasslands. University of Oklahoma Press, Norman.

Gruell, G. E. 1980. Fire's influence on wildlife habitat on the Bridger-Teton National Forest, Wyoming, Volume II- changes and causes, management implications. USDA Forest Service Research Paper INT-252, Ogden, Utah.

Harniss, R. O. and R. B. Murray. 1973. 30 years of vegetal change following burning of sagebrush-grass range. Journal of Range Management 29: 167-168.

Hobbs, R. J. and L. F. Huenneke. 1992. Disturbance, diversity and invasion: implications for conservation. Conservation Biology 6: 324-337.

Hoffman, T. L. 1996. An ecological investigation of mountain big sagebrush in the Gardiner Basin. M. S. Thesis, Montana State University, Bozeman.

Humphrey, L. D. 1984. Patterns and mechanisms of plant succession after fire on *Artemisia*-grass sites in southeastern Idaho. Vegetatio 57: 91-101.

Johnson, J. R. and G. F. Payne. 1968. Sagebrush reinvasion as affected by some environmental influences. Journal of Range Management 21: 209-213.

Klebenow, D. A. 1969. Sage grouse nesting and brood habitat in Idaho. Journal of Wildlife Management 33: 649-661.

Klebenow, D. A. 1973. The habitat requirements of sage grouse and the role of fire in management. Proceedings of the Tall Timbers Fire Ecology Conference 12: 305-315.

Klebenow, D. A. and G. M. Gray. 1967. Food habits of juvenile sage grouse. Journal of Range Management 21: 80-83.

Kotanen, P. M., J. Bergelson and D. L. Hazlett. 1998. Habitats of native and exotic plants in Colorado shortgrass steppe: a comparative approach. Canadian Journal of Botany 76: 664-672.

Lesica, P., S. V. Cooper and G. Kudray. 2005. Big sagebrush shrub-steppe postfire succession in southwest Montana. Report to Bureau of Land Management, Dillon Field Office. Montana Natural Heritage Program, Helena, MT. 29 pp. plus appendices. http://mtnhp.org/community/Reports/Sage_Succession.pdf

McCune, B. and D. Keon. 2002. Equations for potential annual direct incident radiation and heat load. Journal of Vegetation Science 13: 603-606.

Morris, M. S., R. G. Kelsey and D. Griggs. 1976. The geographic and ecological distribution of big sagebrush and other woody *Artemisia*s in Montana. Proceedings of the Montana Academy of Sciences 36: 56-79.

Mueller-Dombois, D. and H. Ellenberg. 1974. Aims and methods of vegetation ecology. John Wiley & Sons, New York.

Nesser, J. A., G. L. Ford, C. L. Maynard and D. S. Page-Dumroese. 1997. Ecological units of the Northern Region: subsections. General Technical Report INT-GTR-369. Ogden, UT: U. S. Department of Agriculture, Forest Service, Intermountain Research Station. 88 pp.

Pechanec, J. F., G. Stewart, and J. P. Blaisdell. 1954. Sagebrush burning-good and bad. U.S. Dep. Agr. Farmer's Bull. 148. 34 p.

Peek, J. M., R. A. Riggs and J. L. Lauer. 1979. Evaluation of fall burning on bighorn sheep winter range. Journal of Range Management 32: 430-432.

Peterson, J. G. 1970. The food habits and summer distribution of juvenile sage grouse in central Montana. Journal of Wildlife Management 34: 147-155.

Pyle, W. H. and J. A. Crawford. 1996. Availability of foods for sage grouse chicks following prescribed fire in sagebrush-bitterbrush. Journal of Range Management 49: 320-324.

Roscoe, J. W. 2002. Sage grouse movements in southwestern Montana. Intermountain Journal of Science 8: 94-104.

Stubbendieck, J., Nichols, James T., and Kelly K. Roberts. 1985. Nebraska range and pasture grasses (including grass-like plants). E.C. 85-170. Lincoln, NE: University of Nebraska, Department of Agriculture, Cooperative Extension Service. 75 p.

Thornton, P. E., S. W. Running and M. A. White. 1997. Generating surfaces of daily meteorological variables over large regions of complex terrain. Journal of Hydrology 190: 214-251.

Walhof, K. S. 1997. A comparison of burned and unburned big sagebrush communities in southwest Montana. Thesis, Montana State University, Bozeman, Montana, USA.

Wallestad, R. 1971. Summer movements and habitat use by sage grouse broods in central Montana. Journal of Wildlife Management 35: 129-135.

Wallestad, R. and D. Pyrah. 1974. Movement and nesting of sage grouse hens in central Montana. Journal of Wildlife Management 38: 630-637.

Wambolt, C. L. and G. F. Payne. 1986. An 18-year comparison of control methods for Wyoming big sagebrush in southwestern Montana. Journal of Range Management 39: 314-319.

Wambolt, C. L., K. S. Walhof and M. R. Frisina. 2001. Recovery of big sagebrush communities after burning in south-western Montana. Journal of Environmental Management 61: 243-252.

Watts, M. J. and C. L. Wambolt. 1996. Long-term recovery of Wyoming big sagebrush after four treatments. Journal of Environmental Management 46: 95-102.

Welch, B. L. and C. Criddle. 2003. Countering misinformation concerning big sagebrush. United States Department of Agriculture, Forest Service Research Paper RMRS-RP-40.

West, N. E. 2000. Synecology and disturbance regimes of sagebrush steppe ecosystems. Pages 15-26 in P. G. Entwistle et al., compilers. Proceedings: sagebrush steppe ecosystems symposium. USDI Bureau of Land Management Publication No. BLM/ID/PT-001001+1150. Boise, Idaho.

West, N. E. and M. A. Hassan. 1985. Recovery of sagebrush-grass vegetation following wildfire. Journal of Range Management 38: 131-134.

Whisenant, Steven G. 1990. Postfire population dynamics of *Bromus japonicus*. American Midland Naturalist. 123: 301-308.

Winward, A. H. 1991. A renewed commitment to management of sagebrush grasslands. Pages 2-7 *in* Research in rangeland management. Agricultural Experiment Station Special Report 880 Oregon State University, Corvallis, Oregon, USA.

Winward, A. H. 2004. Sagebrush of Colorado: Taxonomy, distribution, ecology and management. Colorado Division of Wildlife, Department of Natural Resources, Denver. 45 pp.

Wright, H. A., L. F. Neuenschwander and C. M. Britton. 1979. The role and use of fire in sagebrush-grass and pinyon-juniper plant communities: A state-of-the-art review. USDA General Technical Report INT-58. Ogden, Utah.

Young, J. A., and R. A. Evans. 1989. Dispersal and germination of big sagebrush (*Artemisia tridentata*) seeds. Weed Science 37: 201-206.

Appendix A. Species list of vascular plants that occurred in macroplots

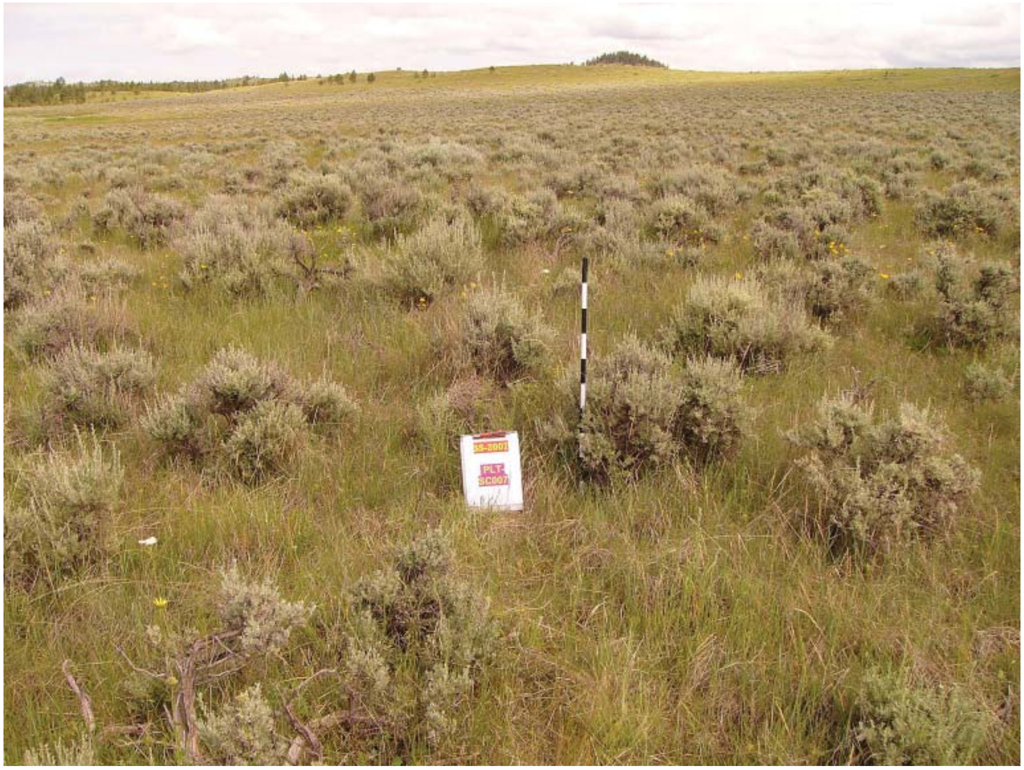
Species list of vascular plants that occurred in macroplots; arranged alphabetically within lifeform; constancy and average cover (%, only for plots in which sp. occurred); not stratified by burn vs. control			
Latin Binomial	**Common Name***	**Constancy**	**Average cover (%)**
SHRUBS			
Artemisia cana	Silver sagebrush	0.17	4.7
Artemisia tridentata ssp. *wyomingensis*	Wyoming big sagebrush	0.65	16.4
Chrysothamnus viscidiflorus	Green rabbitbrush	0.04	0.23
Ericameria nauseosa	Rubber rabbitbrush	0.04	0.34
Juniperus scopulorum	Rocky Mountain juniper	0.02	0.13
Prunus virginiana	Chokecherry	0.02	0.005
Rhus trilobata	Skunkbush sumac	0.02	0.2
Rosa acicularis	Prickly rose	0.02	0.4
Rosa arkansana	Prairie rose	0.02	0.1
Symphoricarpos occidentalis	Western snowberry	0.04	1.4
Tetradymia canescens	Spineless horsebrush	0.10	0.17
SUBSHRUBS			
Artemisia dracunculus	Terragon	0.04	0.1
Artemisia frigida	Prairie sagewort	0.75	1.4
Atriplex gardneri	Gardner’s saltbush	0.04	0.3
Coryphantha vivipara	Pincushion cactus	0.04	0.02
Gutierrezia sarothrae	Broom snakeweed	0.21	0.26
Krascheninnikovia lanata	Winterfat	0.08	0.27
Opuntia fragilis	Brittle pricklypear	0.23	0.12
Opuntia polyacantha	Plains pricklypear	0.54	0.98
Yucca glauca	Soapweed yucca	0.02	0.3
GRAMINOIDS			
Achnatherum hymenoides	Indian ricegrass	0.02	0.05
Agropyron cristatum	Crested wheatgrass	0.08	15.2
Aristida purpurascens	Arrowfeather threeawn	0.02	0.15
Bouteloua gracilis	Blue grama	0.81	7.2
Bromus arvensis (*japonicus*)	Field brome	0.90	15.2
Bromus inermis	Smooth brome	0.06	1.16
Bromus tectorum	Cheatgrass	0.15	0.9
Calamagrostis montanensis	Plains reedgrass	0.02	0.05
Calamovilfa longifolia	Prairie sandreed	0.04	1.2
Carex duriuscula (*stenophylla*)	Needleleaf sedge	0.06	2.1
Carex filifolia	Threadleaf sedge	0.23	1.36
Carex inops ssp. *heliophila*	Sun sedge	0.35	2.9
Danthonia unispicata	Onespike danthonia	0.04	0.3
Elymus elymoides	Squirreltail	0.02	0.05
Elymus (*Agropyron*) *lanceolatus* (*dasystachyum*)	Thickspike wheatgrass	0.08	1.9
Festuca idahoensis	Idaho fescue	0.08	44.4

Hesperostipa (*Stipa*) *comata*	Needle-and-thread	0.60	5.4
Juncus spp.	Rush spp.	0.02	0.1
Koeleria macrantha	Prairie junegrass	0.81	3.1
Nassella (*Stipa*) *virdula*	Green needlegrass	0.63	4.8
Pacopyrum (*Agropyron*) *smithii*	Western wheatgrass	0.96	30.2
Poa pratensis	Kentucky bluegrass	0.19	4.65
Poa secunda	Sandberg's bluegrass	0.98	1.4
Pseudoroegneria (*Agropyron*) *spicata*	Bluebunch wheatgrass	0.35	4.1
Vulpia (*Festuca*) *octoflora*	Sixweeks fescue	0.44	0.8
FORBS			
Achillea millefolium	Common yarrow	0.54	0.46
Agoseris glauca	Pale agoseris	0.08	0.16
Allium textile	Textile onion	0.58	0.09
Alyssum alyssoides	Pale madwort	0.75	0.02
Androsace septentrionalis	Pygmyflower rockjasmine	0.48	0.15
Antennaria neglecta	Field pussytoes	0.35	0.24
Arabis holboellii	Holboell's rockcress	0.04	0.05
Arabis nuttallii	Nuttall's rockcress	0.02	0.05
Arnica sororia	Twin arnica	0.10	1.72
Artemisia ludoviciana	White sagebrush	0.06	0. 6
Asclepias spp.	Milkweed	0.02	0.05
Astragalus adsurgens	Prairie milkvetch	0.02	0.1
Astragalus agrestis	Purple milkvetch	0.27	0.21
Astragalus drummondii	Drummond's milkvetch	0.06	0.7
Astragalus plattensis	Platt River milkvetch	0.02	0.7
Astragalus spp.	Milkvetch spp.	0.02	0.1
Bessia wyomingensis	Wyoming besseya	0.06	0.36
Borage species	Borage spp.	0.06	0.27
Brassicaeae spp.	Mustards	0.13	0.02
Calochortus nuttalliana	Sego lily	0.15	0.03
Camelina microphylla	Littlepod false flax	0.60	0.25
Cerastium arvense	Field chickweed	0.13	0.71
Chamaesyce (*Euphorbia*) *serpylifolia*	Thymeleaf sandmat	0.15	0.05
Chenopodium album	Lambsquarters	0.02	0.01
Cirsium undulatum	Wavyleaf thistle	0.02	0.05
Collinsia parviflora	Maiden blue-eyed Mary	0.06	0.04
Collomia linearis	Narrowleaf blue-eyed Mary	0.23	0.17
Comandra umbellata	Pale bastard toadflax	0.25	0.18
Conyza canadensis	Canadian horseweed	0.06	0.07
Crepis accuminata	Tapertip hawksbeard	0.02	0.25
Crepis intermedia	Limestone hawksbeard	0.04	0.07
Crepis occidentalis	Largeflower hawksbeard	0.08	0.34
Crepis spp.	Hawksbeard spp.	0.15	0.11

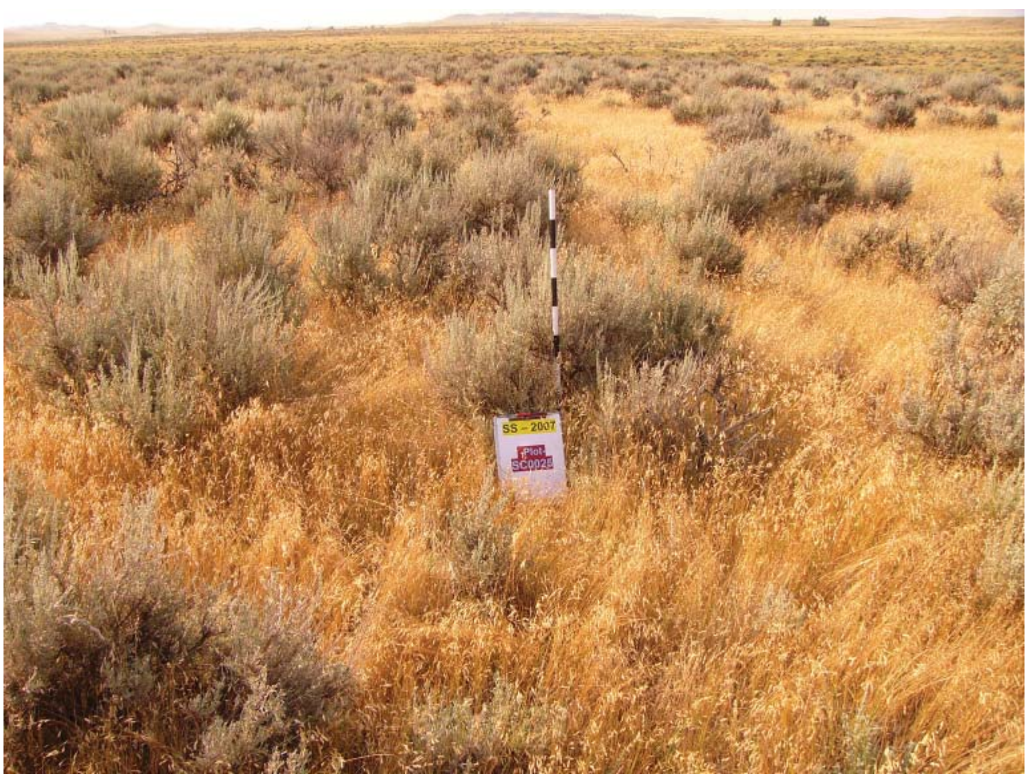
Cryptantha celosioides	Buttecandle	0.13	0.07
Dalea purpurea	Purple prairie clover	0.06	0.18
Descurainia sophia	Herb sophia	0.31	0.16
Draba nemoralis	Eggleaf lacefern	0.04	0.03
Draba oligosperma	Fewseed draba	0.06	0.12
Echinacea angustifolia	Blacksamson echinacea	0.06	2.68
Epilobium paniculatum.	Tall annual willowherb	0.02	0.005
Epilobium spp.	Willowherb spp.	0.02	0.01
Erigeron caespitosus	Tufted fleabane	0.04	0.3
Erigeron pumilus	Navajo fleabane	0.17	0.07
Erigeron spp.	Fleabane spp.	0.02	0.07
Erigeron strigosus	Prairie fleabane	0.04	0.6
Eriogonum spp.	Buckwheat	0.02	0.25
Erysimum repandrum	Spreading wallflower	0.13	0.05
Euphorbia esula	Leafy spurge	0.02	0.09
Fritillaria pudica	Yellow fritillary	0.02	0.01
Galium aparine	Stickywily	0.02	1
Gaura coccinea	Scarlet beeblossom	0.13	0.06
Geum triflorum	Old man's whiskers	0.04	0.22
Hedeoma hispidula	Rough false pennyroyal	0.44	0.06
Helianthus annuus	Common sunflower	0.06	0.06
Heterotheca (*Chrysopsis*) *villosa*	Hairy false goldenaster	0.08	0.1
Heuchera parviflora	Littleflower alumroot	0.02	0.2
Hymonoxysis richardsonii	Pingue rubberweed	0.06	0.1
Ipomoxis aggregata	Scarlet gilia	0.06	0.02
Lactuca seriola	Prickly lettuce	0.21	0.31
Lactuca spp.	Lettuce spp.	0.10	0.09
Lappula occidentalis (*redowskii*)	Flatspine stickseed	0.15	0.06
Lewisia rediviva	Bitter root	0.04	0.26
Liatris punctata	Dotted blazing star	0.23	0.14
Linum lewisii	Lewis flax	0.08	0.14
Linum rigidum	Stiffstem flax	0.08	0.05
Lithospermum incisum	Narrowleaf stoneseed	0.04	0.07
Lithospermum ruderale	Western stoneseed	0.02	0.05
Logfia (*Filago*) *arvensis*	Field cottonrose	0.75	1.27
Lomatium cous	Cous biscuitroot	0.35	0.38
Lomatium orientale	Northern Idaho biscuitroot	0.04	0.07
Lupinus argenteus	Silvery lupine	0.02	0.5
Macaranthera canescens	Hoary tansyaster	0.04	0.1
Medicago sativa	Alfalfa	0.33	0.07
Melilotus officinalis	Yellow sweetclover	0.10	0.6
Mertensia oblongifolia	Oblongleaf bluebells	0.13	0.08
Microseris nutans	Nodding microceris	0.06	0.17

Microsteris gracilis	Slender phlox	0.21	0.19
Musineon divaricatum	Leafy wildparsley	0.13	0.17
Nothocalais troximoides	Weevil prairie-dandelion	0.04	0.08
Oenothera caespitosa	Tufted evening-primrose	0.02	0.05
Oligoneuron rigidum	Stiff goldenrod	0.04	0.07
Orthocarpus luteus	Yellow owl's-clover	0.13	0.17
Oxytropis lagopus	Haresfoot locoweed	0.13	0.06
Oxytropis spp.	Locoweed spp.	0.08	0.15
Paronychia pulvinata	Rocky Mountain nailwort	0.04	0.12
Pediomelum argophyllum	Silverleaf Indian breadroot	0.42	0.27
Pediomelum hypogaeum	Subterranean Indian breadroot	0.02	0.05
Penstemon nitidus	Waxleaf penstemon	0.02	0.2
Penstemon spp.	Beardtongue spp.	0.17	0.13
Phacelia linearis	Threadleaf phacelia	0.06	0.1
Phlox hoodii	Spiny phlox	0.56	0.48
Picradeniopsis oppositifolia	Oppositeleaf bahia	0.10	0.71
Plantago major	Common plantain	0.02	0.05
Plantago patagonica	Woolly plantain	0.77	0.18
Pleiacanthus (*Stephanomeria*) *spinosus*	Thorn skeletonweed	0.02	0.15
Potentilla spp.	Cinquefoil	0.02	0.05
Psoralidium tenuiflorum	Slimflower scurfpea	0.15	0.74
Ratibida columnifera	Upright prairie coneflower	0.08	0.09
Selaginella densa	Lesser spikemoss	0.19	16.4
Silene antirrhina	Sleepy silene	0.04	0.03
Silene spp.	Catchfly spp.	0.02	0.005
Solidago spp.	Goldenrod spp.	0.13	0.23
Sphaeralcea coccinea	Scarlet globemallow	0.79	0.71
Stellaria spp.	Starwort spp.	0.02	0.21
Stenotus (*Haplopappus*) *acaulis*	Stemless mock goldenweed	0.04	0.2
Stephanomeria runcinata	Desert wirelettuce	0.06	0.13
Symphyotrichum falcatum var. *falcatum*	White prairie aster	0.06	0.11
Taraxacum officinale	Common dandelion	0.73	0.58
Tetraneuris (*Hymonoxysis*) *acaulis*	Stemless four-nerve daisy	0.04	0.82
Thermopsis rhombifolia	Prairie thermopsis	0.06	0.27
Tradescantia spp.	Spiderwort	0.02	0.01
Tragopogon dubius	Yellow salsify	0.73	0.2
Veronica arvensis	Corn speedwell	0.06	8.18
Vicia americana	American vetch	0.60	0.63
Viola nuttalliana	Nuttall's violet	0.10	0.22
Zigadenus paniculatus	Foothill deathcamas	0.04	0.05
Zigadenus venosus	Meadow deathcamas	0.23	0.25
* Common & Scientific names according to Natural Resources Conservation Service (USDA) "PLANTS" Database			

Appendix B. Representative Photographs



This plot pair represents the control (top) and burn (bottom) macroplots on a productive, relatively high-elevation site (Diamond Butte vicinity) where the control site is dominated by Wyoming big sagebrush / Idaho fescue – western wheatgrass community type. Wyoming big sagebrush canopy cover on control macroplot is 15%; there is no Wyoming big sagebrush recovery 14 years following prescribed fire. Fescue cover (bottom; note old, tawny stems) decreased in cover whereas western wheatgrass cover more than doubled following fire.



This relatively dry site is characterized by a Wyoming big sagebrush / western wheatgrass – blue grama community type on the control site (bottom). The cover of Wyoming big sagebrush is 22% in the control plot and zero in the burned (wildfire 8 years previously); the dominant grass on both plots is field brome (Japanese brome). The cover of western wheatgrass showed no change with burning, but blue grama cover has increased 20-fold.